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The Safe Urban Harvests Study: A Community-Driven Cross-Sectional Assessment of Metals in Soil, Irrigation Water, and Produce from Urban Farms and Gardens in Baltimore, Maryland

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Background: Emerging evidence suggests social, health, environmental, and economic benefits of urban agriculture (UA). However, limited work has characterized the risks from metal contaminant exposures faced by urban growers and consumers of urban-grown produce.

Objectives: We aimed to answer community-driven questions about the safety of UA and the consumption of urban-grown produce by measuring concentrations of nine metals in the soil, irrigation water, and urban-grown produce across urban farms and gardens in Baltimore, Maryland.

Methods: We measured concentrations of 6 nonessential [arsenic (As), barium (Ba), cadmium (Cd), chromium (Cr), lead (Pb), nickel (Ni)] and three essential [copper (Cu), manganese (Mn), zinc (Zn)] metals in soil, irrigation water, and 13 types of urban-grown produce collected from 104 UA sites. We compared measured concentrations to existing public health guidelines and analyzed relationships between urban soil and produce concentrations. In the absence of guidelines for metals in produce, we compared metals concentrations in urban-grown produce with those in produce purchased from farmers markets and grocery stores (both conventionally grown and U.S. Department of Agriculture–certified organic).

Results: Mean concentrations of all measured metals in irrigation water were below public health guidelines. Mean concentrations of nonessential metals in growing area soils were below public health guidelines for Ba, Cd, Pb, and Ni and at or below background for As and Cr. Though we observed a few statistically significant differences in concentrations between urban and nonurban produce items for some combinations, no consistent or discernable patterns emerged.

Discussion: Screening soils for heavy metals is a critical best practice for urban growers. Given limitations in existing public health guidelines for metals in soil, irrigation water, and produce, additional exposure assessment is necessary to quantify potential human health risks associated with exposure to nonessential metals when engaging in UA and consuming urban-grown produce. Conversely, the potential health benefits of consuming essential metals in urban-grown produce also merit further research. https://doi.org/10.1289/EHP9431

Introduction

Urban agriculture (UA), which includes growing food for human consumption, fiber, plant-based dyes, and other crops, and in some cases, raising livestock and bees, is increasing in popularity worldwide for its numerous public health, community, environmental, and economic benefits (Allen et al. 2008; Santo et al. 2016). Urban agriculture may also help achieve various targets established as part the United Nations' Sustainable Development Goals (Nicholls et al. 2020). In addition, a growing body of literature uses an environmental justice lens to investigate soil contamination concerns in UA (Horst et al. 2017; Malone 2021; McClintock 2012).

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Urban soils may be contaminated with heavy metals [e.g., lead (Pb), arsenic (As), cadmium (Cd)] due to current and historical industrial activities (e.g., fossil fuel combustion, waste incineration, chromate processing) (Harvey et al. 2017), legacy uses of Pb-based paint and leaded gasoline (Mielke et al. 1983, 2011; Mielke and Reagan 1998; Schwarz et al. 2012; Yesilonis et al. 2008), and applications of pesticides (e.g., lead arsenate) (Hood 2006; McBride et al. 2015) as well as their natural occurrence. Urban growers may be exposed to metals via incidental ingestion and inhalation of soil particles and via dermal contact with soil. Consumers may also be exposed via ingestion of contaminated urban-grown produce. A patchwork body of literature exists investigating the relationship between metals in soils and various types of urban-grown produce via uptake (Codling et al. 2016; Codling and Onyeador 2017; McBride et al. 2013, 2015) and other environmental processes (McBride et al. 2014). To date, much of this literature has been focused on a narrow set of metals (e.g., Pb, As) in areas known to be contaminated (Ramirez-Andreotta et al. 2013a, 2013b), resulting in limited generalizability for areas where previous evidence of contamination is absent and in a lack of clear, practicable recommendations for urban growers. As a result, urban growers may encounter unclear or inconsistent recommendations regarding best practices, and consumers may perceive that urban-grown produce is less safe than produce grown in other production systems (Kim et al. 2014). Globally, prior investigations of soil contamination (such as in backyard gardens in Australia (Harvey et al. 2018; Rouillon et al.

2017; Taylor et al. 2021); urban gardens in Spain (De Miguel et al. 2017; Izquierdo et al. 2015); and community gardens in New York, New York (USA) (McBride et al. 2014; Mitchell et al. 2014) have indicated wide variation in metals concentrations across geographic contexts. These findings emphasize the importance of site-specific screening and context-specific guidelines for urban agriculture.

Safety determinations are often made by comparing measured concentrations to established public health guidelines. Although the U.S. Environmental Protection Agency (U.S. EPA) has urban gardening guidance specifically for Pb in soils (U.S. EPA Technical Review Workgroup for Lead 2014), comparable guidance for other metals in soil (e.g., As, Cd) is not available. Existing guidelines, which vary by promulgating agency, are generally intended for the clean-up and remediation of brownfields (Jennings 2013); their appropriateness for the UA context and the safety of food grown in soils is unknown. Furthermore, guidelines for metals in produce exist for only a few metals (e.g., Pb, Cd) in some groups of foods (e.g., leafy greens, root vegetables) (Food and Agriculture Organization 1995). These guidelines are not enforceable and are not available for all metals that may be present in fruits and vegetables, making determination of safety difficult without additional exposure and risk assessments.

Baltimore City, Maryland (henceforth, Baltimore), is a postindustrial city in the mid-Atlantic region of the United States with an emerging and vibrant UA scene. The city's government has enacted and passed numerous policies and programs to promote and encourage UA (Baltimore Office of Sustainability 2013, 2014; Halvey et al. 2021). Baltimore also has a robust network of nonprofit organizations that supports urban growers. Given Baltimore's industrial history and legacies of Pb-based paint in homes, several studies have investigated concentrations of metals (primarily Pb) in city soils (Yesilonis et al. 2008), including some backyard garden soils (Mielke et al. 1983; Schwarz et al. 2016). No citywide attempt has been made to concurrently assess the safety of urban agricultural soils and the produce grown in them in Baltimore.

The Safe Urban Harvests study was a community-driven investigation that aimed to answer community questions about the safety of conducting UA in Baltimore. This study was designed, conducted, interpreted, and communicated in cooperation with several community partners who facilitated the gathering of opinions, questions, and concerns from Baltimore growers and consumers. Community partners included Baltimore City Office of Sustainability, Farm Alliance of Baltimore, Parks and People Foundation (and their formerly active Community Greening Resource Network), and University of Maryland Extension—Baltimore City. We measured the concentrations of nine metals [As, barium (Ba), Cd, chromium (Cr), copper (Cu), Pb, manganese (Mn), nickel (Ni), zinc (Zn)] in soil, irrigation water, and urban-grown produce and compared them to existing public health guidelines. To address urban growers' questions, we characterized relationships between metals in soils and urban-grown produce. Finally, to address consumer questions about the safety of urban-grown produce, we compared the concentrations of metals in urban-grown produce with concentrations of the same metals in nonurban-grown produce (e.g., conventionally grown and U.S. Department of Agriculture (USDA)–certified organic produce purchased from grocery stores and farmers markets).

Methods

Community Engagement

The Safe Urban Harvests Study was conducted in partnership with three local urban agricultural support organizations and city government (hereafter, partners). Partners contributed to study conceptualization, provided feedback on study design, reviewed study protocols, and participated in the interpretation and communication of findings. Partners also played a central role in developing the approach and strategy for reporting back site-specific results and communication of citywide findings to the broader Baltimore community. Throughout the conduct of the study, the study team participated in numerous partner-sponsored events to steward relationships among the study team, partners, and urban growers in Baltimore.

Our earlier work suggested that soil contamination was not a concern for many growers in Baltimore (Kim et al. 2014); however, conversations with community partners revealed that urban growers often cited negative perceptions among consumers about the safety of urban-grown produce relative to produce sold in grocery stores and farmers markets. Accordingly, these perceptions cited by the partners and by urban growers formed the basis for a major element of the study: the comparison of urban- and nonurban-grown produce.

Site Recruitment

We recruited and surveyed 104 urban farms and community gardens (hereafter, UA sites). To be eligible for inclusion a UA site must have *a*) been a food-producing community gardens or urban farm; *b*) located within Baltimore City; *c*) been active during the 2016 or 2017 growing seasons; *d*) distributed their produce to more than one family; and *e*) been growing fruits and vegetables within Baltimore City during the 2017 growing season. Briefly, we identified UA sites using databases developed and maintained by community partners and researched UA sites using websites and social media. Snowball sampling, in which participants and the research team forwarded study information to potential participants, supplemented direct recruitment. Through these methods, we identified 125 UA sites eligible for participation. Ninety-five sites (74% of eligible sites) agreed to participate. We did not actively recruit school/educational gardens but did not exclude the 12 that requested to participate. A site representative (≥18 y old) designated as able to speak for the site completed a survey containing questions about the site history, previous contaminant testing, growing practices (including the use of imported soil), and the numbers of participants and volunteers on site. The Johns Hopkins University Bloomberg School of Public Health institutional review board reviewed study protocols and approved this research.

Sample Collection

Soil samples ($n=616$) were collected between April and November 2017. At least one composite surface (≤15 cm) soil sample (composed of 6–12 subsamples evenly spaced across and representing the growing area) was collected at each site using a stainless-steel garden trowel. Soils at this depth were collected in previous studies evaluating the relationship between metals in soil and produce (McBride et al. 2014, 2015) and were deemed to best represent the soils that growers were most likely to contact while directly growing food (Spliethoff et al. 2016).

Given the wide range in size of growing areas across sites (1–420,000 sq. ft; 0.1–390,190 m^2), the number of composite samples collected at each site and the number of subsamples per composite were determined by the size of the growing area. Two composite samples were collected from sites between 30,000 and 59,999 sq. ft. (2,787 and 5,574 m^2), three composites were collected from sites between 60,000 and 119,999 sq. ft. (5,574 and 11,148 m^2), and four composite samples were collected from sites ≥120,000 sq. ft (11,148 m^2). We used a stainless-steel garden trowel to thoroughly mix composite samples in a small plastic bucket. Both the garden trowel and plastic bucket were washed with laboratory tap water between sites. At 10% of sites, we repeated this collection process a second time. We collected two additional single scoops of soil from nongrowing areas at each site: *a*) pathway soil, i.e., soil from areas highly trafficked by growers but not used for growing, such as dirt pathways or soil between raised beds; and *b*) undisturbed soil from areas rarely trafficked by growers to best approximate soil conditions prior to agricultural use. In addition, 16 sites with active, on-site composting each contributed a compost sample of soil-like consistency. We stored all soil samples in 4-oz (118 mL) Whirl-Pak® bags in a −20°C freezer until further processing.

We collected irrigation water samples ($n=114$) from 92 (88%) sites between April and November 2017. Irrigation water samples were collected only at sites that had active and accessible irrigation water sources when the study team was present on site. At sites that had more than one active and accessible irrigation water source, we collected samples from each source. Samples were collected in 30-mL Nalgene® bottles that were acid washed with 10% nitric acid prior to use. For each source, water flowed for 30 s prior to collection of the sample.

Based on data from administered site surveys (Santo et al. 2021), we collected the 13 most frequently mentioned produce items (excluding herbs) grown in Baltimore: three leafy greens (kale, collards, lettuce), four roots and tubers (carrots, beets, sweet potatoes, potatoes), three nightshade fruits (tomatoes, peppers, eggplants), two cucurbits (summer squash, cucumbers), and one legume (string beans, or sugar snap peas if string beans were unavailable). At least one urban-grown produce sample (total $n=248$) was collected at each of 69 (66%) sites between June and November 2017. An additional single scoop of surface soil (≤15 cm depth) was collected immediately adjacent (≤15 cm) to the base of each plant from which we harvested produce.

During each sampling visit, we reviewed the list of produce items of interest with the site representative. If and only if an item was *a*) being grown on site; *b*) available and ready for harvest; and *c*) the site representative agreed, we collected a produce sample. Recognizing that many of the farms and community gardens rely on produce sales for income and/or food security, we compensated the site manager US$ 5.00 for each sample. Although we aimed to collect at least one sample of at least one type of produce from each site, there were a variety of reasons why we did not collect produce samples at every site: *a*) Most notably, all sites were independently operating farms or gardens whose primary objective was not participation in this research. Although we made attempts and outreach to each farm/garden, not every site representative responded to our request to obtain produce; *b*) Some sites were not growing any of the thirteen items under study and could not contribute a sample; and *c*) Some sites were not interested in produce testing and did not provide approval for our study team to collect a sample. At sites where we collected produce, we collected between one and seven types of produce.

To address community questions about the potential risks of consuming urban-grown produce in comparison with nonurban-grown produce, concurrent with urban-grown produce sample collection, we purchased at least 10 samples of each of the 13 produce items from grocery stores and farmers markets in Baltimore. Grocery stores and farmers markets were purposively selected to include neighborhoods of varying socioeconomic status and store types (e.g., supermarket, international market). At grocery stores, we purchased both conventionally grown (hereafter, conventional) and (when available) USDA-certified organic items (hereafter, organic). At farmers markets, we purchased exclusively from vendors representing farms that were *a*) not located within city limits, i.e., not urban (hereafter, peri-urban) and *b*) not USDA certified organic.

Sample Preparation and Analyses

All soil samples were air dried for 3–5 d, sieved (≤2 mm), digested by aqua regia (nitric and hydrochloric acid), and analyzed using inductively coupled plasma–atomic emission spectroscopy (ICP-AES) with Yttrium as an internal standard (PerkinElmer Optima 4300DV; PerkinElmer) for Ba, Cd, Cr, Cu, Mn, Ni, Pb, and Zn and ICP-AES-Hydride for As at the USDA Agricultural Research Service (ARS) Adaptive Cropping Systems Laboratory in Beltsville, Maryland. Ten grams of air dried, sieved (≤2 mm) soil samples were weighed into beakers and placed on hot plates for digestion using a modified aqua regia procedure (McGrath and Cunliffe 1985). Briefly, concentrated nitric acid (5 mL) and concentrated hydrochloric acid (15 mL) were added to the soil in beakers, covered by a watch glass, heated to near boiling, and periodically swirled over a 2-h reflexing period, followed by continued heating to take the soil samples to near dryness. At this point, 20 mL of 3 N hydrochloric acid were added and once again allowed to reflux for 2 h, after which the samples were filtered, with filtrate being collected into a 50-mL volumetric flask and brought to volume with 0.1 N hydrochloric acid.

Detection limits (LOD) for soil analyses ranged from 0.625–25 ppm as follows: As: 0.625; Ba: 25; Cd: 2; Cr: 2; Cu 4; Ni: 5; Mn: 1; Pb: 5; Zn: 5.5 (Excel Table S1). All elemental concentrations below detection limits were imputed as the limit of detection divided by the square root of 2. For quality control and assurance, 20% field duplicates, 10% blanks, 10% in-house soil standard (Christiana loam), and 10% NIST Standard Reference Materials (SRM) soil [National Institute of Standards and Technology (NIST), Gaithersburg, Maryland] alternating between NIST 2709 San Joaquin soil and NIST 2711a Montana were analyzed. Recovery rates of the SRM were between 90% and 110%. Results from field duplicates were averaged together.

Because the aqua regia method did not speciate Cr, a subset of soil samples ($n=10$) with concentrations of total chromium (Cr) ranging from 60 to 800 ppm were speciated by X-ray absorption near-edge structure (XANES) spectroscopy at the Sector 10-BM beamline (Kropf et al. 2010) of the Advance Photon Source (Argonne National Laboratory). Approximately 50 mg of sample was mixed with 5–10 mg of binder and made into a 7-mm pellet using a pellet press. Pellets were then surrounded with a single layer of Kapton® tape on each side and mounted at the Sector 10-BM beamline and prepared for fluorescence measurement using a Vortex fluorescence detector. A Cr foil reference standard was mounted in front on the reference ionization chamber, allowing for simultaneous sample and reference spectra collection per scan. The beamline was prepared for Cr K edge fluorescence measurements by scanning incident X-ray energy using a silicon (Si) double-crystal (111) monochromator. Collected XANES data were then calibrated to the Cr reference foil, averaged, merged, background subtracted, and normalized using the Demeter Athena software package (Ravel and Newville 2005). Linear combination fitting (LCF) was performed to identify Cr phases present in soil samples. LCF was performed in ATHENA for the Cr XANES from 5,975 eV to 6,100 eV. A library of 18 environmentally relevant Cr standards were used to identify potential components present in each sample. Standards were sequentially removed based on statistical improvement of fit. Components contributing less than 5% were removed, followed by refitting with remaining components. The combination of standards resulting in the lowest *R*-factor results, signifying the statistically best fit for the data, for each sample was reported. All spectra concluded to be a significant LCF component are presented in Figure S1.

All irrigation water samples were acidified to 2% with optima-grade nitric acid after collection and refrigerated until analysis. Collected samples were diluted into 2% HNO3 and 0.5% HCl solution. The calibration curves for As, Ba, Cd, Cr, Cu, Mn, Ni, Pb, Zn were built using a standard solution (Multi-element Aqueous CRM, QC Standard 21; VHG Labs). Ten ppb (vol/vol) of internal standard (CPI International) were added to samples and calibration curves to control potential drifts in the signal. Metal concentrations were then measured using inductively coupled plasma–mass spectrometry (ICP-MS, Agilent 7500ce Octopole; Agilent Technologies). LOD for all elements were <1 ppb (Excel Table S1). All element concentrations below LOD were imputed as the detection limit divided by the square root of 2. For quality control and assurance, 10% field duplicates, 10% lab replicates, 10% blanks, and 10% Standard Reference Material (SRM) samples of trace elements in natural water (NIST SRM® 1643e; NIST) were analyzed. Recovery rates of the SRM were between 90% and 110%. Results from field duplicates and laboratory replicates were averaged together.

All produce samples were rinsed with deionized water to remove visible soil, chopped with stainless steel knives on plastic cutting boards, homogenized in a consumer-grade food processor, transferred to a plastic food-grade container, and frozen. All processing equipment and sample vessels were washed with a sodium lauryl sulfate solution between samples. Samples were subsequently lyophilized, ground in a Wiley mill, and microwave digested (Milestone Ethos Up; Milestone Inc.) with nitric acid and hydrogen peroxide. Ba, Cd, Cr, Cu, Mn, Ni, Zn were analyzed using ICP-AES (PerkinElmer Optima 4300DV) using Yttrium as an internal standard, Pb analyzed by ICP-MS (PerkinElmer Elan DRC), and As by ICP-AES-hydride. A Milestone Ethos UP microwave was used to digest the plant samples. One-gram samples were weighed into quartz inserts to which was added 6 mL of trace metal grade nitric acid and 1 mL deionized water, and 1 mL hydrogen peroxide. The inserts were placed inside microwave vessels containing 3 mL hydrogen perioxide and 5 mL deionized water. The samples under pressure were slowly ramped to 120°C, held for 6 min, slowly ramped to 200°C and held for 15 min, then cooled and diluted to 25 mL volume with deionized water. Trace metal grade acids were used for all analyses.

Dry weight LOD ranged from 0.0125 to 3.75 ppm as follows: As: 0.0137; Ba: 1.25; Cd: 0.375; Cr: 0.375; Cu: 1.5; Ni: 0.625; Mn: 3.75; Pb: 0.0125; Zn: 3.75 (Excel Table S1). For quality control and assurance, 20% field duplicates, 10% blanks, and an in-house vegetable sample were used with every digestion run, matching the vegetable being digested. For each vegetable analyzed, one sample was randomly selected for use in all digestions of that vegetable to assure similar consistency in measurements across digestions. In addition, with every run alternating NIST standards were included, either NIST 1515–apple or NIST 1547–peach. Recovery rates of the SRM were between 90% and 110%. Results from field duplicates were averaged together. Produce sample weights were recorded pre- and post homogenization and after freeze drying to assess water loss during processing and freeze drying. The water content of each sample was calculated as follows:

$$Percent\ water\ content = (sample\ weight_{wet} - sample\ weight_{dry}) / sample\ weight_{wet}.$$

Fresh weight concentrations for all produce samples were calculated as follows:

$$Concentration_{fresh\ weight} = Concentration_{dry\ weight} \times [(100 - percent\ water\ content)/100]$$

(U.S. EPA 2018a).

For all produce samples in which the metal concentration was not detected, we imputed the dry weight LOD and calculated the fresh weight concentration using the sample-specific percent water content. We calculated the bioconcentration factor (BCF) for each metal–produce pair as metal concentration in produce dry weight divided by metal concentration in the single scoop of soil collected immediately adjacent to the produce plant, similar to the method reported in Ramirez-Andreotta (2013a).

Data Analysis

We used Kruskal-Wallis tests to assess differences in metals concentrations between types of soil samples (growing area mixtures, pathway, and undisturbed soil). We used Mann-Whitney rank sum tests to assess differences in growing area metals concentrations between sites that reported growing in exclusively vs. partially imported media. For the six metals with greater than 50% detection in produce study-wide (As, Ba, Cu, Pb, Mn, Zn), we conducted Mann-Whitney tests to assess differences in median concentrations across categories of produce samples (urban vs. nonurban and then urban vs. peri-urban, urban vs. conventional, and urban vs. USDA certified organic). For each variety of urban-grown produce, we calculated pairwise Pearson correlation coefficients to assess the strength of the linear relationship between produce (fresh weight) and soil metal concentrations and bioconcentration factors (BCFs) to characterize the ratio of produce (dry weight) concentrations to soil concentrations. All data analyses were conducted in STATA (version 14; StataCorp) or Python™ (version 3.6; Python Software Foundation). All data displays were produced in Python (version 3.6).

Regulatory Assessment

Regulatory guidance values for metals in soil can vary by several orders of magnitude according to the jurisdiction and intended land use (Jennings 2013). We were unable to locate federal, state, or local regulatory values specific to agricultural soils. We compared soil concentrations to Maryland Soil Standards for residential land use (Maryland Department of the Environment 2018) and the New York State Soil Cleanup Objectives for residential restricted land use (New York State Department of Environmental Conservation and New York State Department of Health 2006) (Excel Table S2). In addition, for Pb, we compared soil concentrations to the U.S. EPA Technical Working Group Guidance for Gardens (U.S. EPA Technical Review Workgroup for Lead 2014). To our knowledge, Pb is the only metal for which the U.S. EPA provides guidance specific to gardening soils. When the values differed for a single metal, we selected the lower of the two values to be most protective.

There are no regulatory guidelines for metals in irrigation water used for UA. We compared irrigation water concentrations to the U.S. EPA's Drinking Water Standard (either primary or secondary) for each metal (U.S. EPA 2018b).

Because there are no regulations for metals in produce in the United States (Nachman et al. 2017, 2018), we used regulatory levels from World Health Organization's *Codex Alimentarius* (Food and Agriculture Organization 1995), which establishes regulatory levels for Cd and Pb in specific types or groupings of produce. To our knowledge, no regulatory guidance for the remaining metals exist (Excel Table S2).

Community Report Back and Risk Communication

A key community aim of this research was to provide each participating site with all measurements and public health interpretations of those measurements for all samples collected from their site. All sites received a document containing soil and irrigation water results in 2018 (Report S1); site representatives who provided a produce sample received a second report detailing produce results in 2019 (Report S2). Our study team emailed or mailed each report to the site's representative and followed up by email and/or phone per growers' preferences to confirm receipt and answer any additional questions about the information contained in their report(s).

Our approach to report-back and design of the document was guided by input from study partners. Briefly, each report contained an executive summary in text summarizing key findings. Next, we presented metal concentrations in relation to existing public health guidelines (for soil and water) or relative to other samples collected (for produce). Finally, in an appendix, we provided data tables of the actual measured concentrations for each medium and metal. Each report also included maps of sample locations on site, an explanation of public health guidelines used for interpretation, best practices for reducing exposure, and information on sources and health effects for all metals included in the report.

Results

Site Characteristics

Of the 104 sites, 62% (64) were community gardens; 17% (18) were urban farms; and the remaining 21% (22) had educational, donation, therapy, or other missions (Santo et al. 2021). Most site representatives reported growing in a combination of raised beds [71% (72)] and directly in ground [58% (59)]. Most sites [91% (95)] relied partially or exclusively on imported growing media (e.g., soil, compost). A total of 86% (89) of sites relied on municipal water (at least in part) for irrigation.

Metals Concentrations in Soil

Rates of detection in soil were greater than 80% for all metals except Cd, which was detected in less than 1% of samples (Excel Table S1). Mean growing area soil concentrations of Ba, Cd, Cu, Ni, and Zn were below the corresponding public health guidelines at all 104 sites (Figure 1). The mean concentration of As (Excel Table S3) in growing area soils exceeded the Maryland Residential Cleanup Standard (0.68 ppm) at all 104 sites but was below the mean U.S. background soil As level (6.4 ppm) (Smith et al. 2013) at 88 sites (85%). The mean concentration of Mn in growing area soils exceeded the Maryland Residential Cleanup Standard (180 ppm) at 97 sites (97%). The mean concentration of Cr (total) in growing area soils exceeded the New York Soil Cleanup Objective for Cr (III) (36 ppm) at 54 sites (52%). The mean concentration of Pb in growing area soils exceeded both the New York Soil Cleanup Objective and the Maryland Residential Cleanup Standard (400 ppm) at four sites (4%).

At 16 sites where we collected compost samples, Ba, Cd, Ni, and Zn concentrations were below the corresponding public health guidelines (for soils) (Excel Table S3). Fourteen compost samples exceeded the Maryland Residential Cleanup Standard for Mn (180 ppm) and 11 for As (0.68 ppm). Three compost samples exceeded the New York Soil Cleanup Objective for Cr (III) (36 ppm) and 1 for Cu (270 ppm). One compost sample exceeded the New York Soil Cleanup Objective for Pb (400 ppm).

Mean concentrations of As, Ba, and Pb were highest in undisturbed soils; and Cu and Mn were highest in growing area soils. Concentrations of Cd, Cr (III), Ni, and Zn were similar across growing area mixtures, pathway, and undisturbed soils (Excel Table S3). Mean concentrations of As, Cr (III), Cu, Ni, and Zn in growing area soils were lower at sites that reported growing in exclusively imported soils in comparison with sites that reported growing in some or no imported media (Excel Table S4).

Cr speciation in 10 soil samples by LCF of XANES spectra (Excel Table S5) showed that Cr (III) was the dominant form present as either chromite ($FeCr_2O_4$) or Cr $(OH)_3$. One sample, VG3, observed about 17% Cr (VI) oxide. Figure S1 shows five Cr standards used for final LCF results to obtain the lowest *R*-factor values; however, Pb chromate and Cr (III) oxide accounted for less than 5% and were not included in the final results despite improving fit results. XANES spectra of the soil samples and corresponding LCF fits are shown in Figure S2.

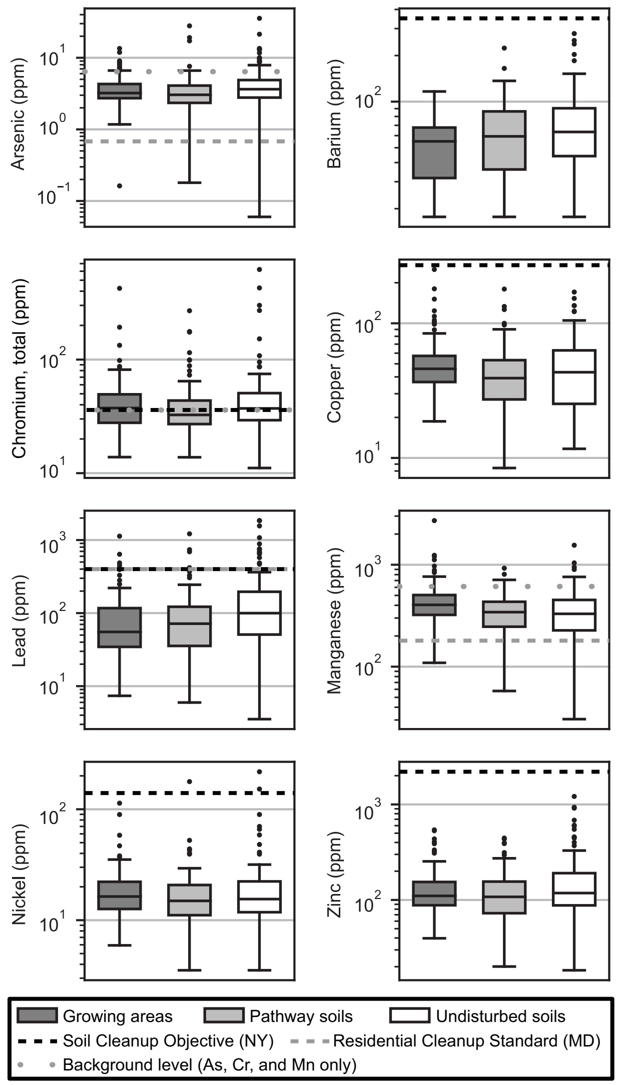

Figure 1. Distribution of mean metal soil concentrations at each urban agriculture site ($n = 104$) by type (growing areas, pathways, undisturbed) relative to public health guidelines. Boxes represent the interquartile range ($IQR = Q3 - Q1$). Upper whiskers extend to the largest number less than $Q3 + 1.5 \times (IQR)$, and lower whiskers extend to the smallest number greater than $Q1 - 1.5 \times (IQR)$. Metals in soil were measured using aqua regia digestion and inductively coupled plasma-atomic emission spectroscopy. Measured values below the limit of detection were imputed as the limit of detection divided by the square root of 2. Summary data are presented in Excel Table S3. Public health guideline values and citations are presented in Excel Table S2. When the residential cleanup standard and soil cleanup objective value differed for a single metal (e.g., As and Cr), we selected and present the lower of the two values. The residential cleanup standard and soil clean-up objective for Pb are both 400 ppm. Background levels for As (6.4 ppm), Cr (36 ppm), and Mn (612 ppm) are from Smith et al. (2013). Note: As, arsenic; Cr, chromium; Mn, manganese; Pb, lead.

Metals Concentrations in Irrigation Water

Rates of detection in irrigation water were greater than 80% for Ba, Cu, Mn, and Zn, and less than 40% for all other metals

(Excel Table S1). Mean concentrations of As, Ba, Cd, Cr, Cu, Ni, and Zn were lower than the drinking water standard for all sources at all sites (Figure S3; Excel Table S6). Mean Pb concentrations from rain-barrel sources at three sites and from municipal sources at two sites were greater than the U.S. EPA drinking water action level (15 ppb) (U.S. EPA 2021). At two sites, the mean concentration of Mn in municipal and rain-barrel sources exceeded the U.S. EPA secondary maximum contaminant limit (50 ppb) (U.S. EPA 2018b) (Figure S3; Excel Table S6).

Concentrations of Metals in Produce Items

Overall, rates of detection were less than 50% for three nonessential [Cd, Cr (total), Ni] metals in produce items (Excel Table S1 and S7), so no further analyses for these metals were conducted. Three nonessential metals (As, Ba, Pb) had rates of detection greater than 50% in produce items (Excel Table S1). Six items (beans, beets, peppers potato, squash, tomato) had less than 50% detection for As (Excel Table S7). Four items (eggplants, pepper, potato, tomato) had less than 50% detection for Ba. All thirteen produce items had at least 50% detection for Pb. All 13 items had at least 50% detection for three essential metals (Cu, Mn, Zn).

Mean and standard deviations (SD) (mean $\pm$ SD) of nonessential metals in all produce items (in decreasing order) were: Ba: $1{,}114 \pm 2{,}283$ ppb; Ni: 82 ± 68 ppb; Pb: 68 ± 97 ppb; Cd: 42 ± 24 ppb; Cr (total) 42 ± 23 ppb; and As: 5 ± 9 ppb (Excel Table S8). Mean concentrations of essential metals in all produce items were: Zn: $2{,}676 \pm 2{,}098$ ppb; Mn: $2{,}160 \pm 2{,}610$ ppb; Cu: $805 \pm 1{,}463$ ppb. Additional descriptive statistics (including range, median, multiple percentiles, and sample sizes) by produce item, production category, and metal are presented in Excel Table S8.

No leafy greens or legumes exceeded the *Codex* guideline for Cd (Excel Table S9). Six root and tuber samples (two organic; two urban; one conventional; one peri-urban) exceeded the *Codex* guideline of 100 ppb for Cd, and two urban-grown fruiting vegetables exceeded the *Codex* guideline of 50 ppb Cd. A total of 107 (58%) root and tuber samples (30 conventional; 27 organic; 26 urban; 24 peri-urban) exceeded the 100 ppb *Codex* guideline for Pb. Fewer than 7% of samples among fruiting vegetables [12 (7%)], legumes [1 (2%)], or leafy greens [1 (1%)] exceeded their guidelines of 100 ppb, 200 ppb, and 300 ppb, respectively.

Comparisons between Urban-Grown and Nonurban-Grown Produce

We assessed differences between urban-grown and nonurban-grown items for all metal–produce combinations (Figure 2). Because we imputed the dry weight detection limit for all samples in which a given metal was not detected before converting to fresh weight, we discuss and highlight only those combinations in which the detection rate was greater than 50% because any differences in concentrations may be attributable to variation in water content, not measured concentrations.

Among nonessential metals, As concentrations in urban-grown produce items (hereafter, urban items) were higher ($p < 0.05$) than those in all other nonurban-grown items (i.e., conventionally grown, certified organic, peri-urban; hereafter, nonurban items) for all three leafy greens (collards, kale, lettuce) (Figure 3). Ba concentrations (Figure S4) in urban items were greater ($p < 0.05$) than those in nonurban items for all three leafy greens (collards, kale, lettuce) and two root vegetables (beets, carrots). Pb concentrations in urban items were greater ($p < 0.05$) for all three leafy greens (collards, kale, lettuce), cucumbers, and peppers (Figure 4).

Among essential metals, Cu concentrations (Figure S5) for urban items were higher ($p < 0.05$) than those in nonurban items for three root vegetables (beets, carrots, potato), three nightshade fruits (eggplant, pepper, tomato), beans, and lettuce. Mn concentrations (Figure S6) in urban items were lower ($p < 0.05$) than those in nonurban items for beans, cucumber, kale, potatoes, squash, and sweet potatoes, but higher ($p = .00002$) for tomatoes. Zn concentrations (Figure S7) in urban items were greater ($p < 0.05$) than those for nonurban items for three root vegetables (beets, carrots, potato), two leafy greens (collards, lettuce), beans, cucumber, lettuce, and three nightshade fruits (eggplant, pepper tomato). Zn concentrations in urban-grown sweet potatoes were less ($p = 0.01$) than those in nonurban items.

Visual inspection of the produce concentration data for all metals demonstrates consistent overlapping of the metal concentration ranges across all four categories of production (urban, conventionally grown, certified organic, peri-urban) (Figures 3–4; Figures S4–S10). It is worth noting, that for some metal–item combinations in which we observed higher median levels of metals in urban items relative to nonurban items overall, we did not necessarily observe higher median levels of the metal in urban items compared with medians in each of the three nonurban production categories. For example, although we observed higher levels of As in urban kale in comparison with nonurban kale overall, we observed higher levels of As in urban kale in comparison with peri-urban kale, but we did not observe higher levels of As in urban kale in comparison with conventional or organic kale. In addition, for some metal–item combinations in which we did not observe higher levels in urban items relative to nonurban items overall, we observed higher levels in urban items relative to one or two (but not all three) of the production categories. For example, although we did not observe higher levels of Pb in urban squash in comparison with nonurban squash, we did observe higher levels of Pb in urban squash relative to organic squash, but not higher levels relative to conventional or peri-urban squash.

Relationship between Urban Soils and Urban-Grown Produce

We presented results only for the item–metal pairs where the detection rate was >50%. Among urban-grown produce items, we did not observe statistically significant linear relationships between soil and produce (fresh weight) concentrations for seven metals: As, Ba, Cd, Cr, Pb, Ni and Zn ($p > 0.05$) (Figure S10). A statistically significant linear relationship between soil and produce concentrations was observed for two essential metals: Cu (rho $= 0.15$, $p < 0.02$) and Mn (rho $= -0.16$, $p < 0.02$) (Excel Table S10).

Mean bioconcentration factors (BCFs) for each metal were all less than 0.3, with the highest mean BCFs observed for Ba (0.28 ± 0.44) and Zn (0.26 ± 0.19). The lowest mean BCFs were observed for Cr (0.01 ± 0.01) and Pb (0.01 ± 0.03) (Excel Table S11). Leafy greens were the produce group with the highest mean BCF for five metals (As, Ba, Cd, Mn, Zn) (Excel Table S11).

Discussion

With rare exception, our findings suggest engaging in UA is safe with respect to metals exposures for urban growers and consumers of urban-grown produce in Baltimore. Concentrations of metals were highest in soils, followed by produce and irrigation water. Mean concentrations of all measured metals in irrigation water were below drinking water guidelines (Figure S3; Excel Table S6). This finding is consistent with the reality that most sites relied on municipal water at least in part as a source of irrigation water. Mean concentrations of nonessential metals in growing area soils were below public health guidelines for Ba, Cd, Pb, and Ni and at or below background for As and Cr (Excel Table S4).

Produce group	Produce item	Production systems compared	Non-essential metals: Arsenic	Non-essential metals: Barium	Non-essential metals: Lead	Essential metals: Copper	Essential metals: Manganese	Essential metals: Zinc
Curcubits	Cucumber	Urban to non-urban	+2.6 (29%)	+27 (22%)	+5 (34%)	-3 (-1%)	-175 (-46%)	+289 (22%)
		Urban to conventional	+0.3 (3%)	+58 (48%)	+13 (86%)	+7 (2%)	-347 (-91%)	+325 (24%)
		Urban to organic	+2.7 (30%)	+41 (34%)	+5 (32%)	-11 (-3%)	-166 (-44%)	+281 (21%)
		Urban to peri-urban	+4.8 (53%)	+7 (6%)	+3 (23%)	+6 (2%)	-131 (-34%)	+195 (15%)
	Squash	Urban to non-urban	+0.3 (28%)	+3 (2%)	+3 (22%)	+10 (2%)	-700 (-107%)	+46 (2%)
		Urban to conventional	+0.3 (32%)	+36 (25%)	+4 (32%)	+77 (14%)	-721 (-110%)	+49 (2%)
		Urban to organic	+0.3 (24%)	-3 (-2%)	+10 (79%)	-73 (-14%)	-716 (-109%)	+67 (3%)
		Urban to peri-urban	+0.3 (27%)	-44 (-31%)	-2 (-14%)	+98 (18%)	-507 (-77%)	+35 (2%)
Leafy greens	Collard	Urban to non-urban	+8.3 (69%)	+1572 (48%)	+22 (49%)	-25 (-6%)	+174 (5%)	+2113 (47%)
		Urban to conventional	+5.4 (45%)	+2572 (79%)	-3 (-7%)	-134 (-33%)	+708 (19%)	+2113 (47%)
		Urban to organic	+7.7 (64%)	+1441 (44%)	+8 (18%)	-54 (-13%)	+564 (15%)	+1996 (44%)
		Urban to peri-urban	+8.7 (72%)	-346 (-11%)	+31 (69%)	+50 (12%)	-2094 (-55%)	+2192 (49%)
	Kale	Urban to non-urban	+2.1 (26%)	+1384 (51%)	+21 (83%)	+50 (8%)	-2142 (-61%)	+587 (14%)
		Urban to conventional	-5.1 (-63%)	+2206 (82%)	+11 (42%)	-146 (-24%)	-2105 (-60%)	-1074 (-25%)
		Urban to organic	-0.6 (-8%)	+1445 (54%)	+22 (86%)	+82 (13%)	-1900 (-54%)	+1810 (42%)
		Urban to peri-urban	+4.8 (59%)	+277 (10%)	+21 (83%)	+101 (16%)	-2180 (-62%)	+556 (13%)
	Lettuce	Urban to non-urban	+1.5 (39%)	+303 (65%)	+23 (73%)	+317 (47%)	+507 (21%)	+1978 (52%)
		Urban to conventional	+1.0 (27%)	+303 (65%)	+22 (68%)	+379 (56%)	+816 (34%)	+1388 (36%)
		Urban to organic	+2.2 (59%)	+375 (80%)	+29 (93%)	+323 (48%)	+869 (37%)	+2219 (58%)
		Urban to peri-urban	+0.7 (18%)	+116 (25%)	+20 (63%)	+136 (20%)	-648 (-27%)	+1803 (47%)
Legume	Bean	Urban to non-urban	+0.6 (35%)	-50 (-11%)	+6 (19%)	+440 (43%)	-479 (-32%)	+1169 (31%)
		Urban to conventional	+0.6 (36%)	+236 (51%)	-2 (-6%)	+479 (47%)	-503 (-34%)	+1180 (32%)
		Urban to organic	+0.6 (33%)	-87 (-19%)	+11 (33%)	+467 (45%)	-174 (-12%)	+1209 (33%)
		Urban to peri-urban	+0.1 (8%)	-267 (-57%)	+6 (18%)	+274 (27%)	-575 (-39%)	+133 (4%)
Nightshade fruits	Eggplant	Urban to non-urban	+0.8 (21%)	+25 (21%)	+12 (21%)	+123 (18%)	-109 (-13%)	+300 (21%)
		Urban to conventional	+0.4 (11%)	+29 (25%)	+28 (51%)	+222 (32%)	-195 (-23%)	+300 (21%)
		Urban to organic	-0.1 (-2%)	+35 (30%)	+12 (22%)	+117 (17%)	+41 (5%)	+410 (29%)
		Urban to peri-urban	+2.1 (58%)	-9 (-8%)	-2 (-4%)	+17 (2%)	-310 (-37%)	+158 (11%)
	Pepper	Urban to non-urban	+0.2 (20%)	+15 (16%)	+2 (6%)	+175 (22%)	-78 (-12%)	+716 (34%)
		Urban to conventional	+0.3 (26%)	+20 (22%)	+31 (95%)	+260 (33%)	-75 (-11%)	+892 (43%)
		Urban to organic	-0.1 (-8%)	+14 (15%)	+14 (42%)	+202 (26%)	+34 (5%)	+747 (36%)
		Urban to peri-urban	+0.2 (20%)	+9 (10%)	-11 (-34%)	+73 (9%)	-275 (-41%)	+311 (15%)
	Tomato	Urban to non-urban	+0.4 (35%)	+45 (41%)	+1 (5%)	+397 (58%)	+215 (29%)	+763 (51%)
		Urban to conventional	+0.4 (39%)	+48 (44%)	+17 (57%)	+512 (75%)	+285 (38%)	+934 (62%)
		Urban to organic	+0.4 (34%)	+44 (41%)	-1 (-3%)	+406 (60%)	+185 (25%)	+751 (50%)
		Urban to peri-urban	+0.2 (21%)	+26 (24%)	+8 (27%)	+271 (40%)	+107 (14%)	+648 (43%)
Roots and tubers	Beet	Urban to non-urban	+0.5 (20%)	+1845 (55%)	-32 (-43%)	+479 (40%)	-450 (-24%)	+1980 (42%)
		Urban to conventional	+0.4 (18%)	+2750 (81%)	-21 (-28%)	+482 (40%)	-374 (-20%)	+803 (17%)
		Urban to organic	+0.4 (16%)	+1734 (51%)	-32 (-43%)	+446 (37%)	-472 (-25%)	+2872 (61%)
		Urban to peri-urban	+0.6 (24%)	+1422 (42%)	-66 (-89%)	+444 (37%)	-1128 (-59%)	+1942 (41%)
	Carrot	Urban to non-urban	-2.2 (-83%)	+1262 (63%)	+95 (39%)	+471 (48%)	-24 (-3%)	+1607 (48%)
		Urban to conventional	-2.7 (-102%)	+1326 (66%)	+68 (28%)	+474 (49%)	+52 (6%)	+1881 (56%)
		Urban to organic	-4.2 (-157%)	+1344 (67%)	+126 (52%)	+507 (52%)	-40 (-4%)	+1837 (55%)
		Urban to peri-urban	+0.8 (30%)	+287 (14%)	+117 (48%)	+287 (30%)	-380 (-42%)	+1278 (38%)
	Potato	Urban to non-urban	-0.2 (-9%)	-13 (-5%)	-27 (-42%)	+544 (36%)	-303 (-33%)	+1342 (36%)
		Urban to conventional	-0.3 (-11%)	-25 (-10%)	-44 (-67%)	+833 (56%)	-374 (-41%)	+1581 (43%)
		Urban to organic	-0.4 (-16%)	-24 (-10%)	-42 (-65%)	+448 (30%)	-63 (-7%)	+1274 (34%)
		Urban to peri-urban	-0.0 (-0%)	+8 (3%)	+5 (7%)	+456 (30%)	-558 (-62%)	+620 (17%)
	Swt. potato	Urban to non-urban	+2.6 (37%)	-223 (-30%)	-173 (-328%)	+187 (13%)	-926 (-69%)	-579 (-29%)
		Urban to conventional	+3.1 (45%)	-642 (-85%)	-244 (-463%)	+141 (10%)	-916 (-68%)	-764 (-38%)
		Urban to organic	-0.3 (-5%)	-124 (-16%)	-161 (-306%)	+308 (21%)	-885 (-66%)	-391 (-20%)
		Urban to peri-urban	+2.7 (39%)	-236 (-31%)	-144 (-274%)	-28 (-2%)	-1727 (-128%)	-844 (-42%)

Difference (%)	Urban >75% higher than non-urban, P<0.05
Difference (%)	Urban 25-75% higher than non-urban, P<0.05
Difference (%)	No statistically significant difference between urban and non-urban
Difference (%)	Detection rate for item and metal <50%
Difference (%)	Urban >25% lower than non-urban, P<0.05

Figure 2. Comparisons of metals concentrations in urban and nonurban produce, reported as absolute (ppb) and relative (percent) differences in median values. Absolute differences (e.g., median of metal in urban-grown produce minus median of metal in conventional produce) and percent differences in medians were calculated. We used two-sided Mann-Whitney-U tests ($p < 0.05$) to assess differences in median concentrations across categories of produce samples (urban vs. nonurban and then urban vs. peri-urban, urban vs. conventional, and urban vs. organic). The number of samples in each group is found in Excel Table S9. Nonurban includes peri-urban, conventional, and organic. Note: ppb, parts per billion.

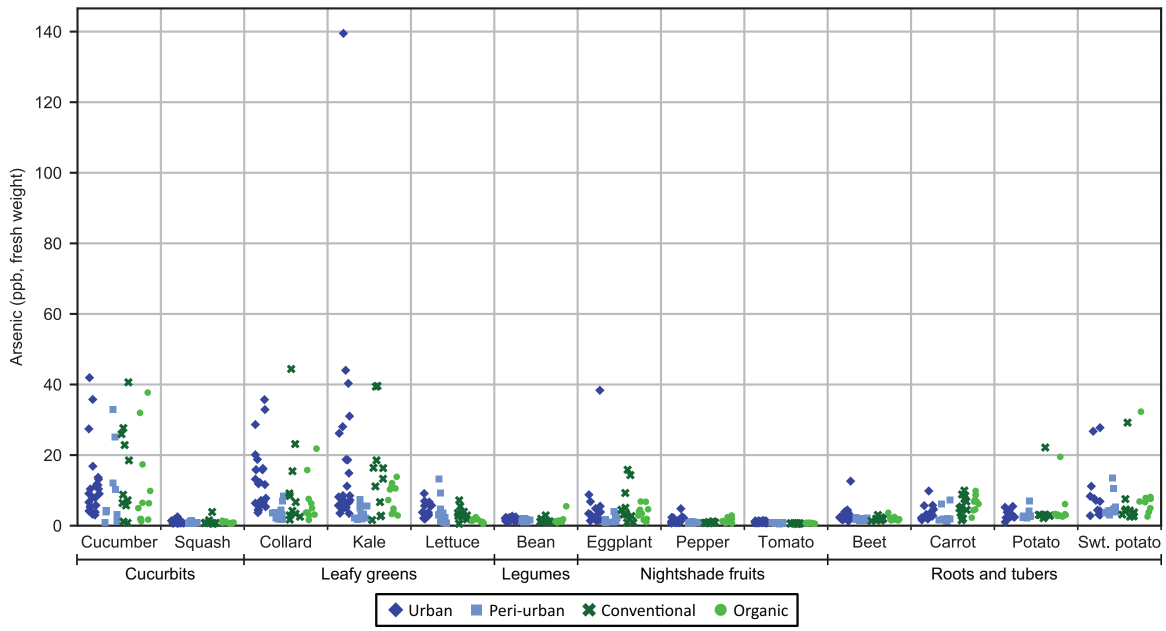

Figure 3. Fresh weight arsenic concentrations (ppb) measured in produce items, by production category. Arsenic in produce was measured using inductively coupled plasma–atomic emission spectroscopy-hydride. Measured values below the dry weight limit of detection were imputed as the limit of detection divided by the square root of 2. All dry weight concentrations were converted to fresh weight using sample-specific water content. Summary data are presented in Excel Table S8. Note: ppb, parts per billion.

Based on our comparisons of the concentrations of metals in urban- and nonurban-grown produce (Figure 2), our main finding for produce consumers is that we found no reason to recommend changes in produce consumption, either by type of produce or by production system (e.g., urban, peri-urban, conventional, or organic.) Although we observed a few statistically significant differences in metals concentrations between urban- and nonurban-grown produce for some produce items, we found no discernable pattern in these differences. Any observed differences are likely an artifact of the multiple comparisons made.

We observed variation in metals concentrations across vegetable types (Figures 3–4 and Figures S4–10). We observed the highest levels of Pb in root vegetables, regardless of production category (Excel Table S8). We observed higher levels of As, Ba, and Pb in urban-grown leafy greens in comparison with nonurban-grown (Figure 2), though almost all Pb concentrations were far below the World Health Organization's recommendations for Pb levels in leafy greens (Excel Table S9). Although these absolute differences may be informative, additional modeling and assessment are needed to determine whether these

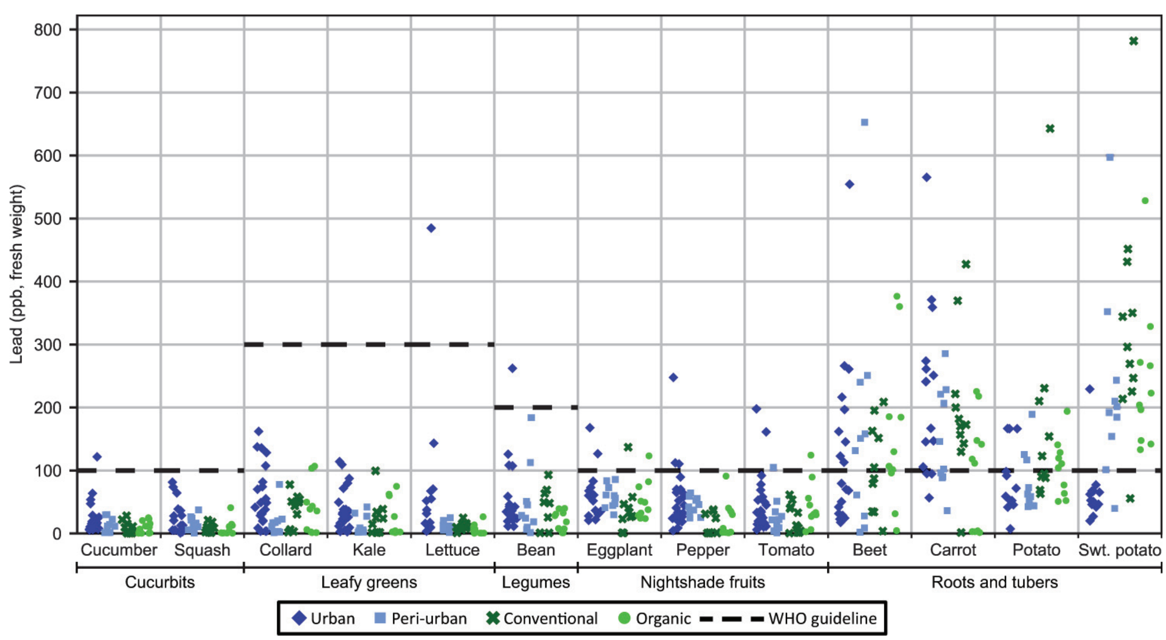

Figure 4. Fresh weight lead concentrations (ppb) measured in produce items, by production category. Lead in produce was measured using inductively coupled plasma–mass spectrometry. Measured values below the dry weight LOD were imputed as the LOD divided by the square root of 2. All dry weight concentrations were converted to fresh weight using sample-specific water content. Summary data are presented in Excel Table S8. Note: LOD, limit of detection; ppb, parts per billion.

differences translate into appreciable increased risks or benefits for consumers. Moreover, although we present concentrations of metals in different produce groups (Figures 3–4 and Figures S4–10) (e.g., nightshade fruits, root vegetables, leafy greens), our investigation was not designed to compare the safety of different kinds of fruits and vegetables to each other; further analyses must also consider differences in typical amounts consumed of each item.

We did not observe significant correlations between levels of metals in urban soils and urban-grown produce (Figure S11 and Excel Table S10), suggesting that a higher concentration of metals in soil is not a strong indicator of higher concentrations in produce grown in that soil and that determinations of safety for produce and soil should be made independently. This finding was consistent for all 13 produce items investigated. Across urban soils, we generally observed higher concentrations of metals in nongrowing areas, e.g., pathways and undisturbed areas (Excel Table S4 and S5). This finding may suggest that growers' practices (e.g., importing soil and applying soil amendments) may dilute and/or reduce the concentrations of contaminants in soils (Moskal and Berthrong 2018), though we did not collect enough information about site-specific growing practices (e.g., types and amounts of amendments applied and at what frequency) to evaluate this hypothesis. Although our research contributes to the overall body of literature characterizing the relationship between metals in soil and produce, additional field-based experiments are needed to further characterize and quantify the human (e.g., growing and irrigation practices), ecological (e.g., weather conditions, soil health parameters), and plant-specific (e.g., cultivar) factors that may affect direct uptake of metals.

Although existing public health guidelines (Excel Table S2) are an important decision-making tool for public health practitioners and urban growers, our findings suggest the need for a broader discussion and more nuanced application of such guidelines. For example, applying the U.S. EPA's Soil Screening Level for As (0.68 ppm) as the sole decision-making tool in this study would have led to a determination that all 104 participating agriculture sites were not "safe" for production due to As levels in soil; however, this guideline as derived is lower than the naturally occurring background level of As in soils and is not intended to suggest the site is not suitable for the safe conduct of UA when exceeded. Rather, this guideline, when exceeded, suggests additional investigation of the metal exposure may be warranted via a site-specific risk assessment. Furthermore, this soil guideline has no connection to or consideration of levels of As in produce, so application of this guideline in isolation would not address questions about the safety of consuming produce grown in soils.

Using the U.S. EPA's guideline for Pb in soils as the sole decision-making tool for determination of site safety would have made four UA sites "unsafe" for UA; however, in practice, 35 of these sites yielded at least one produce sample that exceeded *Codex* guidelines. Furthermore, we must acknowledge that these guidelines are generally not intended for the UA context, so the assumptions about exposure used to derive such guidelines may not be truly representative of exposures in UA. For example, the 400 ppm guideline for Pb in soil, although widely used and cited as an acceptable level for Pb in soils, was derived in 1994 by the U.S. EPA for the specific purpose of protecting children from unacceptable exposures to Pb in residential soils and preventing 95% of the population of U.S. children from having blood Pb levels exceeding 10 $\mu g/dL$, which was the Centers for Disease Control and Prevention (CDC) action level for blood Pb at the time. Given increasing evidence that there is no level of Pb known to be without adverse effect in children, the CDC has reduced the blood Pb reference value to 5 $\mu g/dL$ (Centers for Disease Control and Prevention 2021). Questions remain, however, regarding whether the current guideline is adequately protective for children (Gilbert and Weiss 2006; Gottesfeld 2021; Paulson and Brown 2019). Although childhood and pregnancy are vulnerable life stages for Pb exposure, children's exposures to soil are likely less and also less relevant for this context than those of agricultural workers and community gardeners who routinely engage in intentional contact with soils while growing food. Guidelines for Pb in soils intended to protect adults engaging in UA should be linked to more appropriate and relevant end points (e.g., cardiovascular, neurodegeneration, and others) for adult life stages. Interventions on Pb exposures are most successful if they address all the potential exposure pathways (e.g., dietary, inhalation, ingestion); in the context of UA, where metals exposures may be greater, critical pieces of information needed to derive and/or revise guidelines for soils are estimates of soil intake for children and adults engaged in agriculture. To date, the U.S. EPA has low confidence in existing estimates of soil intake among children and adults in the general population (U.S. EPA. 2017), and no estimates of soil intake exist for agricultural workers and other populations who have occupational or recreational soil contact.

In addition, these soil guidelines do not consider specific exposure patterns (e.g., time spent on site or specific activities conducted) or exposure-reduction behaviors (e.g., use of tools and/or personal protective equipment such as gloves). For example, more focused investigations of how growers divide their time among specific tasks (e.g., planting, irrigating, weeding) at both a daily and seasonal level would greatly improve our understanding and estimation of soil exposure in the UA context (Lupolt et al. 2021). Although existing exposure models have modeled the transport pathway of outdoor soil and indoor dust (Layton and Beamer 2009), additional investigation is needed to evaluate the relevance of these models to the UA context. Specifically, these models should incorporate the contribution of human interaction with the environment (e.g., urban growers' deliberate and intentional disruption and transportation of soil). Furthermore, guidelines for metals in soil do not account for aggregate exposures that may occur via other pathways, such as inhalation and dietary intake. For urban growers, a single guideline for soil, irrigation water, or produce should not be applied as a single binary determination of the safety of UA at a given site, but as part of a nuanced site-specific assessment, with the limitations and caveats for each guideline noted.

Existing public health guidance for metals in produce adopts one of two approaches: a single food regulatory approach or a total diet guideline approach. The former describes the approach employed by the Food and Agriculture Organization in *Codex Alimentarius* (Food and Agriculture Organization 1995), which establishes regulatory limits for a single food (or food group) and a single metal using estimates of intake for that food and toxicity information about that metal. The latter approach, adopted by the U.S. Food and Drug Administration (U.S. FDA), establishes a daily tolerable intake for Pb in the entire diet for adults and children (Carrington and Bolger 1992). Such a guideline, which to date has only been provisionally established for Pb, is helpful for determining overall risks from dietary exposures, but better surveillance of the metals in the food supply is needed to develop actionable recommendations (by identifying specific foods or groups of foods) for consumers or regulatory agencies interested in reducing exposures.

A key methodological strength of our study was the use of laboratory-based analytical methods for soil and produce that allowed us to achieve low analytical detection limits (Excel Table S1). As other environmental sources of Pb continue to decline (Dignam et al. 2019; Resongles et al. 2021), it is increasingly necessary to quantify the remaining exposures with adequate precision. Thus, a sensitive laboratory method is critical for risk analyses and future decision-making. In addition, we recorded sample weights pre- and post lyophilization, which enabled us to calculate fresh weight concentrations using sample-specific water content, increasing the precision of our produce concentration data. Similar studies have reported metals concentration in dry weight only (Finster et al. 2004) or used standard estimates of water content (Kohrman and Chamberlain 2014; McBride et al. 2014). In addition, we are among the first to present metals concentrations of 13 commonly grown produce items, across 4 production methods, facilitating multiple comparisons by both produce type and production method.

Previous investigations (Mitchell et al. 2014; Spliethoff et al. 2016) of metals in soils used for UA have used similar public health guidelines for decision-making. In the absence of public health guidelines for metals in foods, other researchers have compared metals concentrations to concentrations measured in the U.S. FDA's Total Diet Study (McBride et al. 2014; Ramirez-Andreotta et al. 2013a, 2013b). Other researchers have conducted risk assessments using default assumptions about incidental soil ingestion and/or produce consumption intended for the general population or study-specific assumptions (Entwistle et al. 2019; Hough et al. 2004; Manjón and Ramirez-Andreotta 2020; Ramirez-Andreotta et al. 2013a; Spliethoff et al. 2016). Although these assessments are helpful for assessing site safety in retrospect, their study-specific assumptions make comparison of studies and generalizability about the safety of UA difficult. Furthermore, because these studies were conducted on active UA operations, this approach may limit the transferability of findings to new operations and do not provide consistent characterization of site conditions and practices nor specific exposure pattern information to advance the development of public health guidelines intended for UA growers and consumers.

In recognition of the deficiencies of specific guidelines, additional research is needed to better understand agricultural practices among urban growers, and consumption patterns of urban-grown produce relative to nonurban produce among different populations to derive more applicable and nuanced public health guidelines. Given the limitations of existing public health guidelines and the limitations in their interpretation demonstrated in our study, our findings suggest a paradigm shift is needed in the establishment of public health guidelines that sufficiently account for the nuanced reality of metal exposure—including exposures via other pathways.

Based on our findings, we believe that guidelines for agricultural soils should be derived using estimates of soil ingestion and time in contact with soil specific to growers, rather than the general U.S. population. Guidelines for metals in produce should incorporate a more nuanced and tiered approach that considers exposures that occur through the diet overall as well as targets those individual foods that may contribute the most to dietary exposures. Furthermore, because fruits and vegetables (both urban- and nonurban-grown) are a critical part of a healthy diet and provide important nutritional benefits, a more nuanced risk–benefit model of guidance may provide more actionable recommendations for consumers. Guidance developed jointly by the U.S. FDA and the U.S. EPA for methylmercury in fish (U.S. Food and Drug Administration 2020) may be a helpful model for providing guidance for balancing nonessential metals and beneficial nutrients in a wider array of produce.

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References

Allen JO, Alaimo K, Elam D, Perry E. 2008. Growing vegetables and values: benefits of neighborhood-based community gardens for youth development and nutrition. J Hunger Environ Nutr 3(4):418–439, https://doi.org/10.1080/19320240802529169.

Baltimore Office of Sustainability. 2013. Homegrown Baltimore: Grow Local: Baltimore City's Urban Agriculture Plan. http://www.baltimoresustainability.org/wp-content/uploads/2015/12/HGB-Grow-Local-Final-Cover-1.pdf [accessed 25 March 2018].

Baltimore Office of Sustainability. 2014. *The City of Baltimore's Soil Safety Policy for Food Production*. Baltimore, MD: Baltimore Office of Sustainability.

Carrington CD, Bolger PM. 1992. An assessment of the hazards of lead in food. Regul Toxicol Pharmacol 16(3):265–272, PMID: 1293643, https://doi.org/10.1016/0273-2300(92)90006-u.

Centers for Disease Control and Prevention. 2021. Blood lead reference value. https://www.cdc.gov/nceh/lead/data/blood-lead-reference-value.htm [accessed 17 September 2021].

Codling EE, Chaney RL, Green CE. 2016. Accumulation of lead and arsenic by potato grown on lead–arsenate-contaminated orchard soils. Commun Soil Sci Plant Anal 47(6):799–807, https://doi.org/10.1080/00103624.2016.1146754.

Codling EE, Onyeador J. 2017. Accumulation of lead and arsenic in Malabar spinach (*Basella Alba* L.) and sweet potato (*Ipomoea batatas* L.) leaves grown on urban and orchard soils. J Plant Nutr 40(20):2898–2909, https://doi.org/10.1080/01904167.2017.1382530.

De Miguel E, Izquierdo M, Gómez A, Mingot J, Barrio-Parra F. 2017. Risk assessment from exposure to arsenic, antimony, and selenium in urban gardens (Madrid, Spain). Environ Toxicol Chem 36(2):544–550, PMID: 27467197, https://doi.org/10.1002/etc.3569.

Dignam T, Kaufmann RB, LeStourgeon L, Brown MJ. 2019. Control of lead sources in the United States, 1970–2017: public health progress and current challenges to eliminating lead exposure. J Public Health Manag Pract 25 (suppl 1, Lead Poisoning Prevention):S13–S22, PMID: 30507765, https://doi.org/10.1097/PHH.0000000000000889.

Entwistle JA, Amaibi PM, Dean JR, Deary ME, Medock D, Morton J, et al. 2019. An apple a day? Assessing gardeners' lead exposure in urban agriculture sites to improve the derivation of soil assessment criteria. Environ Int 122:130–141, PMID: 30449630, https://doi.org/10.1016/j.envint.2018.10.054.

Finster ME, Gray KA, Binns HJ. 2004. Lead levels of edibles grown in contaminated residential soils: a field survey. Sci Total Environ 320(2–3):245–257, PMID: 15016510, https://doi.org/10.1016/j.scitotenv.2003.08.009.

Food and Agriculture Organization. 1995. *Codex Alimentarius. General Standard for Contaminants and Toxins in Food and Feed*. Geneva, Switzerland: World Health Organization, Food and Agriculture Organization of the United Nations.

Gilbert SG, Weiss B. 2006. A rationale for lowering the blood lead action level from 10 to 2 μg/dL. Neurotoxicology 27(5):693–701, PMID: 16889836, https://doi.org/10.1016/j.neuro.2006.06.008.

Gottesfeld P. 2021. Finding the next Flint: the need to update the blood lead reference value. Am J Public Health 111(10):1746–1749, PMID: 34499533, https://doi.org/10.2105/AJPH.2021.306429.

Halvey MR, Santo RE, Lupolt SN, Dilka TJ, Kim BF, Bachman GH, et al. 2021. Beyond backyard chickens: a framework for understanding municipal urban agriculture policies in the United States. Food Policy 103:102013, https://doi.org/10.1016/j.foodpol.2020.102013.

Harvey PJ, Peterson PG, Taylor MP. 2018. VegeSafe: a community science program generating a national residential garden soil metal(loid) database. Environ Sci Pollut Res Int 25(33):33745–33754, PMID: 30276696, https://doi.org/10.1007/s11356-018-3293-3.

Harvey PJ, Rouillon M, Dong C, Ettler V, Handley HK, Taylor MP, et al. 2017. Geochemical sources, forms and phases of soil contamination in an industrial city. Sci Total Environ 584–585:505–514, PMID: 28129903, https://doi.org/10.1016/j.scitotenv.2017.01.053.

Hood E. 2006. The apple bites back: claiming old orchards for residential development. Environ Health Perspect 114(8):A470–A476, PMID: 16882511, https://doi.org/10.1289/ehp.114-a470.

Horst M, McClintock N, Hoey L. 2017. The intersection of planning, urban agriculture, and food justice: a review of the literature. J Am Plann Assoc 83(3):277–295, https://doi.org/10.1080/01944363.2017.1322914.

Hough RL, Breward N, Young SD, Crout NM, Tye AM, Moir AM, et al. 2004. Assessing potential risk of heavy metal exposure from consumption of home-produced vegetables by urban populations. Environ Health Perspect 112(2):215–221, PMID: 14754576, https://doi.org/10.1289/ehp.5589.

Izquierdo M, De Miguel E, Ortega MF, Mingot J. 2015. Bioaccessibility of metals and human health risk assessment in community urban gardens. Chemosphere 135:312–318, PMID: 25966050, https://doi.org/10.1016/j.chemosphere.2015.04.079.

Jennings AA. 2013. Analysis of worldwide regulatory guidance values for the most commonly regulated elemental surface soil contamination. J Environ Manag 118:72–95, PMID: 23399880, https://doi.org/10.1016/j.jenvman.2012.12.032.

Kim BF, Poulsen MN, Margulies JD, Dix KL, Palmer AM, Nachman KE. 2014. Urban community gardeners' knowledge and perceptions of soil contaminant risks. PLoS One 9(2):e87913, PMID: 24516570, https://doi.org/10.1371/journal.pone.0087913.

Kohrman H, Chamberlain CP. 2014. Heavy metals in produce from urban farms in the San Francisco Bay area. Food Addit Contam Part B Surveill 7(2):127–134, PMID: 24914598, https://doi.org/10.1080/19393210.2013.859740.

Kropf AJ, Katsoudas J, Chattopadhyay S, Shibata T, Lang EA, Zyryanov VN, et al. 2010. The new MRCAT (sector 10) bending magnet beamline at the advanced photon source. AIP Conference Proceedings 1234:299–302, https://doi.org/10.1063/1.3463194.

Layton DW, Beamer PI. 2009. Migration of contaminated soil and airborne particulates to indoor dust. Environ Sci Technol 43(21):8199–8205, PMID: 19924944, https://doi.org/10.1021/es9003735.

Lupolt SN, Agnew J, Burke TA, Kennedy RD, Nachman KE. 2021. Key considerations for assessing soil ingestion exposures among agricultural workers. J Expo Sci Environ Epidemiol, PMID: 34079062, https://doi.org/10.1038/s41370-021-00339-z.

Malone M. 2021. Seeking justice, eating toxics: overlooked contaminants in urban community gardens. Agric Hum Values, https://doi.org/10.1007/s10460-021-10236-8.

Manjón I, Ramírez-Andreotta M. 2020. A dietary assessment tool to estimate arsenic and cadmium exposures from locally grown foods. Environ Geochem Health 42(7):2121–2135, PMID: 31845019, https://doi.org/10.1007/s10653-019-00486-1.

Maryland Department of the Environment. 2018. Cleanup Standards for Soil and Groundwater. https://mde.maryland.gov/programs/LAND/MarylandBrownfieldVCP/Documents/www.mde.state.md.us/assets/document/MDE%20Soil%20and%20Groundwater%20Cleanup%20Standards%2010-2018%20Interim%20Final%20Update%203-2.pdf [accessed 4 November 2021].

McBride MB, Shayler HA, Russell-Anelli JM, Spliethoff HM, Marquez-Bravo LG. 2015. Arsenic and lead uptake by vegetable crops grown on an old orchard site amended with compost. Water Air Soil Pollut 226(8):265, PMID: 26900187, https://doi.org/10.1007/s11270-015-2529-9.

McBride MB, Shayler HA, Spliethoff HM, Mitchell RG, Marquez-Bravo LG, Ferenz GS, et al. 2014. Concentrations of lead, cadmium and barium in urban garden-grown vegetables: the impact of soil variables. Environ Pollut 194:254–261, PMID: 25163429, https://doi.org/10.1016/j.envpol.2014.07.036.

McBride MB, Simon T, Tam G, Wharton S. 2013. Lead and arsenic uptake by leafy vegetables grown on contaminated soils: effects of mineral and organic amendments. Water Air Soil Pollut 224(1):1378, PMID: 26884640, https://doi.org/10.1007/s11270-012-1378-z.

McClintock N. 2012. Assessing soil lead contamination at multiple scales in Oakland, California: implications for urban agriculture and environmental justice. Appl Geogr 35(1–2):460–473, https://doi.org/10.1016/j.apgeog.2012.10.001.

McGrath SP, Cunliffe CH. 1985. A simplified method for the extraction of the metals Fe, Zn, Cu, Ni, Cd, Pb, Cr, Co and Mn from soils and sewage sludges. J Sci Food Agric 36(9):794–798, https://doi.org/10.1002/jsfa.2740360906.

Mielke HW, Anderson JC, Berry KJ, Mielke PW, Chaney RL, Leech M. 1983. Lead concentrations in inner-city soils as a factor in the child lead problem. Am J Public Health 73(12):1366–1369, PMID: 6638229, https://doi.org/10.2105/ajph.73.12.1366.

Mielke HW, Laidlaw MA, Gonzales CR. 2011. Estimation of leaded (Pb) gasoline's continuing material and health impacts on 90 US urbanized areas. Environ Int 37(1):248–257, PMID: 20825992, https://doi.org/10.1016/j.envint.2010.08.006.

Mielke HW, Reagan PL. 1998. Soil is an important pathway of human lead exposure. Environ Health Perspect 106 (suppl 1):217–229, PMID: 9539015, https://doi.org/10.1289/ehp.98106s1217.

Mitchell RG, Spliethoff HM, Ribaudo LN, Lopp DM, Shayler HA, Marquez-Bravo LG, et al. 2014. Lead (Pb) and other metals in New York City community garden soils: factors influencing contaminant distributions. Environ Pollut 187:162–169, PMID: 24502997, https://doi.org/10.1016/j.envpol.2014.01.007.

Moskal BT, Berthrong ST. 2018. Novel soil barrier systems potentially protect urban growing beds from legacy soil contamination and improve soil health. Urban Agriculture & Regional Food Systems 3(1):1–9, https://doi.org/10.2134/urbanag2018.06.0003.

Nachman KE, Ginsberg GL, Miller MD, Murray CJ, Nigra AE, Pendergrast CB. 2017. Mitigating dietary arsenic exposure: current status in the United States and recommendations for an improved path forward. Sci Total Environ 581–582:221–236, PMID: 28065543, https://doi.org/10.1016/j.scitotenv.2016.12.112.

Nachman KE, Punshon T, Rardin L, Signes-Pastor AJ, Murray CJ, Jackson BP, et al. 2018. Opportunities and challenges for dietary arsenic intervention. Environ Health Perspect 126(8):84503, PMID: 30235424, https://doi.org/10.1289/EHP3997.

New York State Department of Environmental Conservation, New York State Department of Health. 2006. New York state brownfield cleanup program development of soil cleanup objectives: technical support document. https://www.dec.ny.gov/docs/remediation_hudson_pdf/techsuppdoc.pdf [accessed 4 November 2021].

Nicholls E, Ely A, Birkin L, Basu P, Goulson D. 2020. The contribution of small-scale food production in urban areas to the sustainable development goals: a review and case study. Sustain Sci 15(6):1585–1599, https://doi.org/10.1007/s11625-020-00792-z.

Paulson JA, Brown MJ. 2019. The CDC blood lead reference value for children: time for a change. Environ Health 18(1):16, PMID: 30819209, https://doi.org/10.1186/s12940-019-0457-7.

Ramirez-Andreotta MD, Brusseau ML, Artiola JF, Maier RM. 2013a. A greenhouse and field-based study to determine the accumulation of arsenic in common homegrown vegetables grown in mining-affected soils. Sci Total Environ 443:299–306, PMID: 23201696, https://doi.org/10.1016/j.scitotenv.2012.10.095.

Ramirez-Andreotta MD, Brusseau ML, Beamer P, Maier RM. 2013b. Home gardening near a mining site in an arsenic-endemic region of Arizona: assessing arsenic exposure dose and risk via ingestion of home garden vegetables, soils, and water. Sci Total Environ 454–455:373–382, PMID: 23562690, https://doi.org/10.1016/j.scitotenv.2013.02.063.

Ravel B, Newville M. 2005. ATHENA, ARTEMIS, HEPHAESTUS: data analysis for X-ray absorption spectroscopy using IFEFFIT. J Synchrotron Radiat 12(pt 4):537–541, PMID: 15968136, https://doi.org/10.1107/S0909049505012719.

Resongles E, Dietze V, Green DC, Harrison RM, Ochoa-Gonzalez R, Tremper AH, et al. 2021. Strong evidence for the continued contribution of lead deposited during the 20th century to the atmospheric environment in London of today. Proc Natl Acad Sci USA 118(26):e2102791118, PMID: 34155116, https://doi.org/10.1073/pnas.2102791118.

Rouillon M, Harvey PJ, Kristensen LJ, George SG, Taylor MP. 2017. VegeSafe: a community science program measuring soil-metal contamination, evaluating risk and providing advice for safe gardening. Environ Pollut 222:557–566, PMID: 28027776, https://doi.org/10.1016/j.envpol.2016.11.024.

Santo R, Palmer AM, Kim BF. 2016. Vacant Lots to Vibrant Plots: A Review of the Benefits and Limitations of Urban Agriculture. Baltimore, MD: Johns Hopkins Center for a Livable Future.

Santo RE, Lupolt SN, Kim BF, Burrows RA, Evans E, Evenson B, et al. 2021. Characteristics and growing practices of Baltimore city farms and gardens. Urban Forestry & Urban Greening 65:127357.

Schwarz K, Pickett ST, Lathrop RG, Weathers KC, Pouyat RV, Cadenasso ML. 2012. The effects of the urban built environment on the spatial distribution of lead in residential soils. Environ Pollut 163:32–39, PMID: 22325428, https://doi.org/10.1016/j.envpol.2011.12.003.

Schwarz K, Pouyat RV, Yesilonis I. 2016. Legacies of lead in Charm City's soil: lessons from the Baltimore ecosystem study. Int J Environ Res Public Health 13(2):209, PMID: 26861371, https://doi.org/10.3390/ijerph13020209.

Smith D, Cannon WF, Woodruff LG, Solano F, Kilburn JE, Fey D. 2013. Geochemical and Mineralogical Data for Soils of the Conterminous United States: Data Series 801. Denver, CO: U.S. Geological Survey, https://doi.org/10.3133/ds801.

Spliethoff HM, Mitchell RG, Shayler H, Marquez-Bravo LG, Russell-Anelli J, Ferenz G, et al. 2016. Estimated lead (Pb) exposures for a population of urban community gardeners. Environ Geochem Health 38(4):955–971, PMID: 26753554, https://doi.org/10.1007/s10653-016-9790-8.

Taylor MP, Isley CF, Fry KL, Liu X, Gillings MM, Rouillon M, et al. 2021. A citizen science approach to identifying trace metal contamination risks in urban gardens. Environ Int 155:106582, PMID: 33940394, https://doi.org/10.1016/j.envint.2021.106582.

U.S. EPA (U.S. Environmental Protection Agency). 2017. *Exposure Factors Handbook Chapter 5 (Update): Soil and Dust Ingestion. EPA/600/R-17/384F*. Washington, DC: U.S. Environmental Protection Agency.

U.S. EPA. 2018a. *Exposure Factors Handbook Chapter 9 (Update): Intake of Fruits and Vegetables. EPA/600/R-17/384F*. Washington, DC: U.S. Environmental Protection Agency.

U.S. EPA. 2018b. 2018 Edition of the Drinking Water Standards and Health Advisories Tables. https://www.epa.gov/sites/default/files/2018-03/documents/dwtable2018.pdf [accessed 4 November 2021].

U.S. EPA Technical Review Workgroup for Lead. 2014. Technical Review Workgroup Recommendations Regarding Gardening and Reducing Exposure to Lead-Contaminated Soils. Washington, DC: U.S. Environmental Protection Agency.

U.S. EPA. 2021. Lead and Copper Rule. https://www.epa.gov/dwreginfo/lead-and-copper-rule [accessed 4 November 2021].

U.S. Food and Drug Administration. 2020. Advice about eating fish: for women who are or might become pregnant, breastfeeding mothers, and young children. https://www.fda.gov/food/consumers/advice-about-eating-fish [accessed 25 January 2021].

Yesilonis ID, Pouyat RV, Neerchal NK. 2008. Spatial distribution of metals in soils in Baltimore, Maryland: role of native parent material, proximity to major roads, housing age and screening guidelines. Environ Pollut 156(3):723–731, PMID: 18656291, https://doi.org/10.1016/j.envpol.2008.06.010.